Nitrate Leaching from Cattle Urine and Feces in Northeast USA

W. L. Stout,* S. A. Fales, L. D. Muller, R. R. Schnabel, W. E. Priddy, and G. F. Elwinger

ABSTRACT

Management intensive grazing (MIG) is a grazing system in which animals at a high stocking density are rotated frequently through a series of paddocks in a manner that maximizes both forage yield and quality. Although MIG has the potential to increase dairy farm profitability in the U.S. Northeast, the uneven recycling of N through feces and urine can increase NO_3–N leaching. The extent to which NO_3–N can leach from beneath urine and fecal spots under soil and climatic conditions of the Northeast is not known. We conducted a field study to measure NO_3–N leaching loss from spring-, summer-, and fall-applied urine and summer-applied fecal beneath N-fertilized orchardgrass (*Dactylis glomerata* L., cv. Pennlate) using 60-cm-diameter by 90-cm-deep drainage lysimeters. The study site was located in central Pennsylvania on a Hagerstown silt loam soil (fine, mixed, mesic Typic Hapludalf). Averaged across the 3 yr of the study, NO_3–N losses were 1.17, 1.68, 22.0, 24.0, and 31.5 g m^{-2} for the control, feces, and spring-, summer-, and fall-applied urine, respectively. These losses represent about 2% of the N applied in the feces and about 18, 28, and 31% of the spring-, summer-, and fall-applied urine N. If dairy farmers in the Northeast continue to increase the utilization of MIG, the amount of N leached to the groundwater from beneath pastures could become substantial if not mitigated by improved grazing management.

DAIRY FARMING is the largest single agricultural enterprise in the U.S. Northeast, with milk receipts accounting for >25% of the farm income in this region (U.S. Department of Agriculture, 1991). Milk production in this region accounts for about 20% of the milk production nationwide. Thus, maintaining a profitable dairy industry in the Northeast is desirable at both a regional and national level.

To maintain dairy farm profitability in the face of rising fuel and machinery costs, tighter environmental constraints, and decreasing federal subsidies, some researchers and extension specialists in the Northeast are advocating inclusion of MIG as a component of dairy production in this region (Fales et al., 1993). Management intensive grazing is a grazing system in which animals at a high stocking density are rotated frequently through a series of paddocks in a manner that maximizes both forage yield and quality. Management intensive grazing systems have been the mainstay of dairy production in the temperate oceanic climatic zones of Europe and New Zealand for many years, and MIG has the potential to increase farm profitability in the Northeast (Emmick and Toomer, 1991; Parker et al., 1992). However, the higher rates of N used in these grazing systems compared with mechanically harvested forages and the uneven recycling of this N through feces and urine in pastures has increased N leaching, which can pose a threat to water quality (Ball et al., 1979; Ryden et al., 1984; Steenvoorden et al., 1986).

Nitrate leaching from intensively grazed pastures occurs in humid regions where precipitation exceeds evapotranspiration and results from the high levels of N fertilization and the uneven recycling of N in urine and feces (Petersen et al., 1956a). Nitrate leached from grazed grasslands receiving high N inputs far exceeds that leached from similarly fertilized cut grasslands and is affected by soil type and grazing density (Steenvoorden et al., 1986; Garwood and Ryden, 1986). Nitrate-N concentrations averaged 25 mg L^{-1} in groundwater draining an intensive grazing area in New Zealand (Barber and Wilson, 1972).

Grazing density affects NO_3–N loss through leaching, because the bulk of the N consumed by the animal is excreted in the urine as urea (Ball and Ryden, 1984; Jarvis et al., 1989). Depending on temperature, some of this N is volatilized as NH_3 (Harper et al., 1983a,b); however, most of the urea rapidly nitrifies to NO_3 and is subject to leaching or denitrification (Ball et al., 1979).

Urine and fecal spots cover about a 60-cm-diameter area (Petersen et al., 1956b; Garwood and Ryden, 1986), and N concentrations under urine spots are often equal to a 700 kg ha^{-1} fertilizer N application. Nitrate leaching from grazed grasslands has been shown to increase substantially when N application exceeds 425 kg ha^{-1} (Barraclough et al., 1992).

Since MIG has the potential to increase NO_3–N leaching and MIG use is increasing in the Northeast, baseline leaching data needs to be developed for dairy production systems that are both economically and environmentally sustainable. Our objective was to determine the NO_3–N leaching loss from fertilizer N, urine, and feces under the temperate continental climatic conditions common to the northeastern USA.

METHODS AND MATERIALS

The study was conducted at The Pennsylvania State University Dairy Research Center located in central Pennsylvania (40°48′N, 77°52′W, 350 m elev.). The predominant soil on the site is a Hagerstown silt loam. This soil is a deep, well-drained soil formed in relatively pure limestone residuum. Although the subsoil texture of the Hagerstown series is a clay loam, drainage through the subsoil is rapid because of a high degree of well-defined blocky structure (Shuford, 1975).

Our leaching loss study took place in conjunction with a MIG study on this site. During the 3 yr of the study, N application rates on the pastures ranged from 19.6 to 28.0 g m^{-2} (196–280 kg ha^{-1}) as NH_4NO_3, depending on the weather and the number of grazing cycles in a given year (Table 1). The lysimeters were located adjacent to the paddocks of the MIG study, but were protected from being directly impacted by grazing cattle with 3 by 1.25 m steel gates. Both control and treatment lysimeters were fertilized and harvested at the same time as the paddocks were grazed and fertilized (Table 1). The paddocks were managed by turning the animals onto the

W.L. Stout, R.R. Schnabel, W.E. Priddy, and G.F. Elwinger, USDA-ARS Pasture Systems and Watershed Management Research Lab., Curtin Road, University Park, PA 16802-3702; and S.A. Fales, 115 AS&I Building, and L.D. Muller, 316 Henning Building, Pennsylvania State Univ., University Park, PA 16802. Received 26 Aug. 1996. *Corresponding author (wsl@psu.edu).

Published in Soil Sci. Soc. Am. J. 61:1787–1794 (1997).

Table 1. Blanket N fertilization dates and rates for pastures, urine lysimeters, feces lysimeters, and control lysimeters.

1993		1994		1995	
Date	N rate	Date	N rate	Date	N rate
	$g\ N\ m^{-2}$		$g\ N\ m^{-2}$		$g\ N\ m^{-2}$
8 Apr.	5.6	11 Apr.	5.6	11 Apr.	8.4
6 May	5.6	28 Apr.	5.6	1 May	5.6
18 May	5.6	16 May	5.6	9 June	2.8
20 Aug.	5.6	8 June	2.8	12 July	2.8
21 Sept.	5.6	15 June	2.8		
		15 Aug.	2.8		
Total	28.0	Total	25.2	Total	19.6

pasture when the sward height was approximately 30 cm and removing them when the sward height was reduced to 7.5 cm. The lysimeters were managed by cutting the herbage to 7.5 cm with electric grass shears at the same time the paddocks were grazed. Cutting dates were 5 May, 18 May, 15 June, 8 July, 20 August, 21 September, and 26 October in 1993; 28 April, 16 May, 6 June, 6 July, 15 August, 20 September, and 31 October in 1994; and 1 May, 9 June, 12 July, 24 August, and 31 October in 1995.

The monolith lysimeters used in this study were constructed in the winter of 1992–1993 using the design developed by Moyer et al. (1996) at the Rodale Research Farm. First, an intact soil core was taken by driving a 100 by 60 cm section of schedule 40 steel well casing 90 cm into the N-fertilized orchardgrass paddocks before the grazing study started. Ninety centimeters was assumed to be the bottom of the root zone and 60 cm was the diameter of influence of a urine or fecal spot (Petersen et al., 1956a). Next, the core was retrieved and a bottom and collection system were installed onto the bottom of the core. Finally, the core was replaced into the soil. No suction was applied to the bottom of the lysimeters, thus leachate volume may have been lower and denitrification rates may have been higher than under an intact soil column. However, monolith lysimeters of such design generally provide the most effective method of measuring NO_3–N leaching from many types of soils (Whitehead, 1995).

Lysimeter installation was complete in time for the 1993 grazing starting in April. The lysimeters were checked for leachate weekly or after major storm events. Leachate was removed from the lysimeter with a small electric pump, leachate volume was recorded, and subsamples stored under refrigeration at 4°C and filtered with 0.45-μm membrane filters using Millipore apparatus (Millipore Corp., Bedford, MA). The NO_3–N analysis was performed (U.S. Environmental Protection Agency, 1979) using a Waters ILC/1 ion chromatograph (Waters Chromatography Div., Milford, MA) using an Altech 269-013 column with a phthalic acid mobile phase. The analysis was performed as soon as possible after sampling, but in any case, delayed by no more than 30 d.

Urine and feces were collected on the day of treatment or 1 or 2 d prior to treatment from the animals grazing the paddocks while they were in the holding area for milking. If collected prior to treatment, urine was stored at 4°C. A 3-L urine application (Petersen et al., 1956a) was made in the spring, summer, and fall (Table 2) to urine treatment lysimeters, coincident with the paddocks being grazed. A 2-kg feces application (Petersen et al., 1956a) was made to the feces lysimeters only in the summer after a midsummer grazing. The excreta was applied to the center of the urine and feces lysimeters to simulate animal deposition. There were control lysimeters that received no urine or feces, only the blanket N application (Table 1). The same lysimeters received urine or feces each year, received urine or feces at the same time each year, and the same control lysimeters were used every year. There were five replications of each treatment for a total of 25 lysimeters.

Air temperature, precipitation, radiation, relative humidity, and wind velocity were measured on site using two Campbell Scientific weather stations; additional measurements were made off site by The Pennsylvania State University Meteorology Department.

The experiment was analyzed as a randomized complete block using the GLM procedure in SAS (SAS Institute, 1988). Initial analysis indicated that there was a significant year × treatment interaction for leachate volume, NO_3–N concentration, and NO_3–N loss. Consequently, the data were analyzed and presented by year. Since diagnostic procedures indicated that the variances of both the NO_3–N loss and concentration values were correlated with their means, cube root transformations were used to stabilize the variances. Analysis of variance and mean separation procedures were performed on the transformed scales. The use of the word *significant* throughout the text indicates $P < 0.05$.

RESULTS

Weather

Temperature and precipitation data for the 3 yr of the study are summarized in Fig. 1. While the temperature pattern remained fairly constant during the 3 yr of the study, there was quite a bit of variation in the precipitation pattern. In the 1993 and 1994 grazing seasons (i.e., April–October) precipitation was evenly distributed. However, from mid-July to mid-September in the 1995 grazing season, there was <25 mm of precipitation. This is only about 15% of the precipitation that is usually received at this location during this time of the year. Consequently, N fertilization was reduced to 19.6 g m^{-2} in the 1995 grazing season (Table 1).

Nitrogen Application

Nitrogen application rates to lysimeters resulting from excreta applications averaged 92.9 g m^{-2} for urine and 28.4 g m^{-2} for feces (Table 2). The difference in N application between seasons and years was due to the variation in the N concentration in the urine. Urine N concentration reflected the N content of the herbage in the pastures and was highest in the spring or fall. Even

Table 2. Amounts of N applied in urine and feces treatments to lysimeters containing N-fertilized orchardgrass.

	1993		1994		1995	
Treatment	Date	N applied	Date	N applied	Date	N applied
		$g\ N\ m^{-2}$		$g\ N\ m^{-2}$		$g\ N\ m^{-2}$
Urine in spring	18 May	96.6	29 Apr.	112.0	1 May	141.8
Urine in summer	9 July	69.9	14 Aug.	81.2	12 July	95.6
Feces in summer	9 July	28.9	14 Aug.	28.9	12 July	27.3
Urine in fall	21 Sept.	107.9	2 Nov.	103.8	31 Oct.	88.4

at the highest urine or feces N application rate, the grass in the lysimeters was not damaged.

Leachate volumes, NO_3–N concentrations, and total NO_3–N loss are summarized in Table 3. The N application rate from urine is about three times higher than that from feces, but the potential impact of urine on leachate NO_3–N concentration was disproportionally higher for two reasons. Urine immediately infiltrates into the soil where the urea is readily hydrolyzed to NH_3, nitrified, and becomes more subject to leaching than to volatilization. In contrast, feces remains on the surface where organically bound N is subject to volatil-

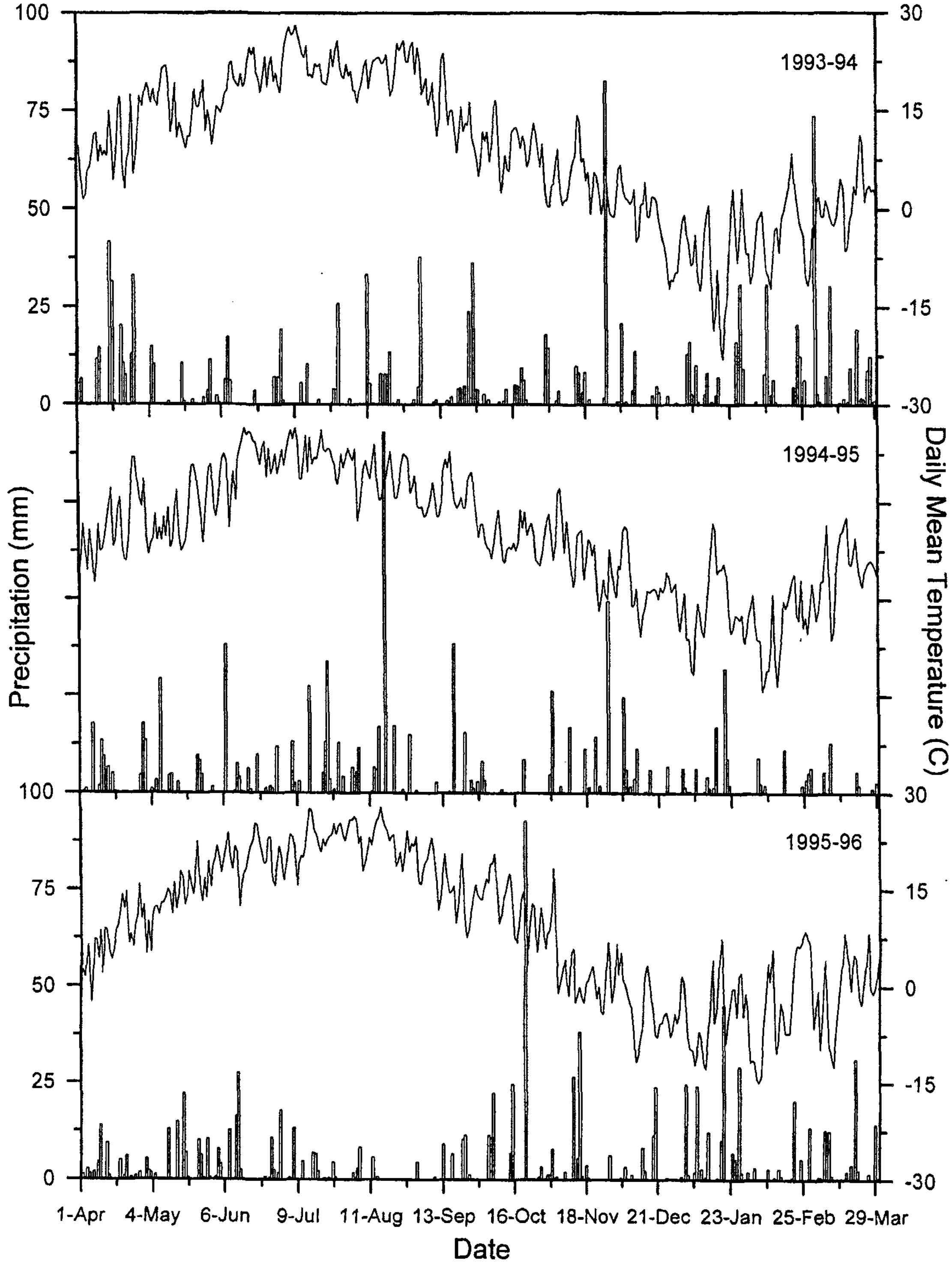

Fig. 1. Daily mean temperature and precipitation at the study site.

Table 3. Summary of leachate amount, NO_3–N concentration, and NO_3–N loss from drainage lysimeters for 3 yr.

Year	Control	Feces Summer	Urine Spring	Urine Summer	Urine Fall
		Leachate amount, mm			
1993	267a†	259a	203a	205a	234a
1994	325a	249ab	215b	264ab	217b
1995	246a	242a	234a	276a	228a
		NO_3–N concentration, mg L^{-1}			
1993	5.0c	5.8c	78.9ab	52.1b	90.7a
1994	4.6c	4.8c	88.9b	76.5b	152a
1995	4.9b	9.0b	164a	151a	176a
		NO_3–N leaching loss, g m^{-2}			
1993	1.16b	1.51b	10.5b	16.1ab	21.5a
1994	1.15c	1.50c	18.9b	14.6b	32.4a
1995	1.19b	2.04b	36.8a	41.4a	40.5a

† Means in the same row followed by the same letter are not significantly different ($P \leq 0.05$). Year × treatment interaction significant ($P \leq 0.05$) for all parameters.

ization as NH_3 is produced during the mineralization process.

Leachate Amount

The leachate patterns were similar for all 3 yr of the study (Table 3, Fig. 2). Leachate volumes were lower during the grazing season when evapotranspiration was the highest and increased in the fall as evapotranspiration decreased.

In 1993 and 1995, there was a clear pattern of leachate flow beginning in the fall, followed by a decrease during the winter months when the ground was frozen, and an increase again after the spring thaw. In the fall of 1994, however, leachate flow remained relatively high throughout the winter and did not decrease until evapotranspiration increased in late spring. During 1994, leachate flow continued into late summer (Fig. 1) due to precipitation and a large storm event of almost 100 mm.

There was a significant year × treatment interaction for leachate amount resulting from leachate amount being significantly higher under the control than under the spring and fall urine treatments in one of the 3 yr (1994). This was due to increased N fertility imparted by the urine treatments enabling the grass growing in the lysimeters to make better use of the increased precipitation in 1994, consequently increasing evapotranspiration. This response was not evident in 1993 and 1995 because there was less precipitation in these years and soil water rather than N limited grass growth and water use.

Nitrate Concentration

The pattern of NO_3–N concentration in leachate from the lysimeters was similar for the 3 yr of the study (Fig.

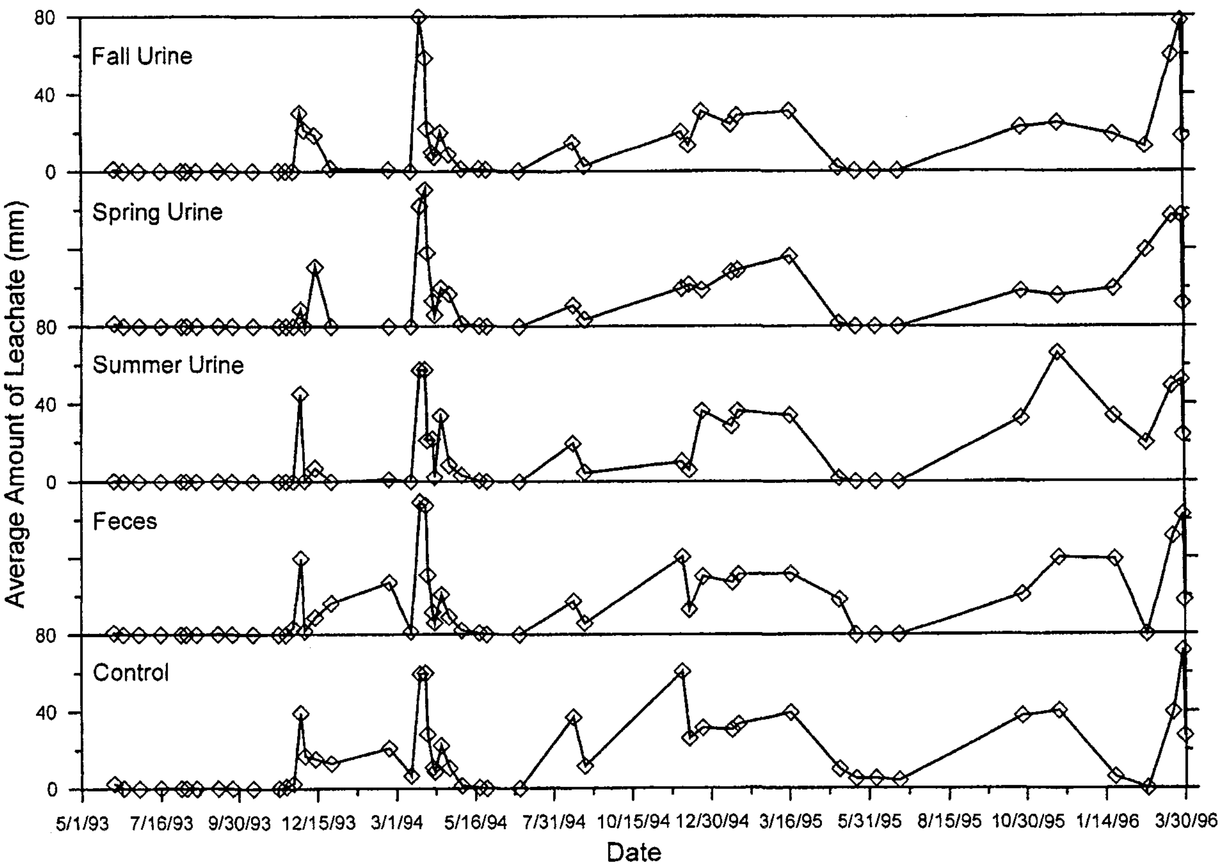

Fig. 2. Average leachate volumes for April 1993 to March 1994, April 1994 to March 1995, and April 1995 to March 1996.

3). Although there was a significant year × treatment interaction for NO_3–N concentration, NO_3–N concentrations beneath fall-applied urine were always highest (Table 3). This was because there was insufficient time left in the growing season for this N to be assimilated by the grass before it was subject to increased leaching in the fall.

Nitrate-N concentrations in leachate beneath spring- and summer-applied urine were generally lower, despite the fact that N application rates under spring-applied urine tended to be higher during the study (Table 2). Lower NO_3–N concentrations beneath spring-applied urine are likely because grass receiving urine in the spring has most of the growing season to assimilate the applied N. The lower NO_3–N concentrations beneath the summer-applied urine can be attributed to higher N volatilization from urine during the summer (Harper et al., 1983b).

In contrast to urine, NO_3–N concentrations in leachate beneath feces were not significantly different from those beneath the control (Fig. 3, Table 3). This was a result of the much lower amount of N applied in the feces than in urine, the potential volatilization of N from the feces (Ryden, 1986), and the lower availability of N in feces. Nitrate-N concentrations beneath all treatments were greatest in 1995, the driest year of the study when N assimilation by the grass was limited by a lack of soil moisture.

Nitrate-Nitrogen Leaching Loss

In general most NO_3–N was leached beneath the fall-applied urine in all years of the study (Table 3, Fig. 4). However, there was a significant treatment × year interaction caused by NO_3–N leaching beneath spring- and summer-applied urine being equal to that under fall-applied urine in 1 yr of the study. This was due to low precipitation (Fig. 1) during the middle of the grazing season in 1995, which retarded grass growth and N assimilation. Consequently, more N was available for leaching from spring- and summer-applied urine in 1995 than in the other 2 yr.

The leaching of NO_3–N from beneath the feces was not significantly different than that from beneath the control (Fig. 4, Table 3). During the 3 yr of the study, the additional NO_3–N leaching loss (i.e., treatment minus control) that could be attributed to feces-applied N was 0.51 g m^{-2} or about 2% of the N applied in the feces. In contrast, additional NO_3–N leaching losses that could be attributed to spring-, summer-, and fall-applied urine were 20.9, 22.9 and 30.3 g m^{-2}, respectively. These amounts represent 18, 28, and 31% of the spring-, summer-, and fall-applied urine N, respectively.

DISCUSSION

The observations that grazed grasslands have higher N leaching losses than cut grasslands (Ryden et al., 1984)

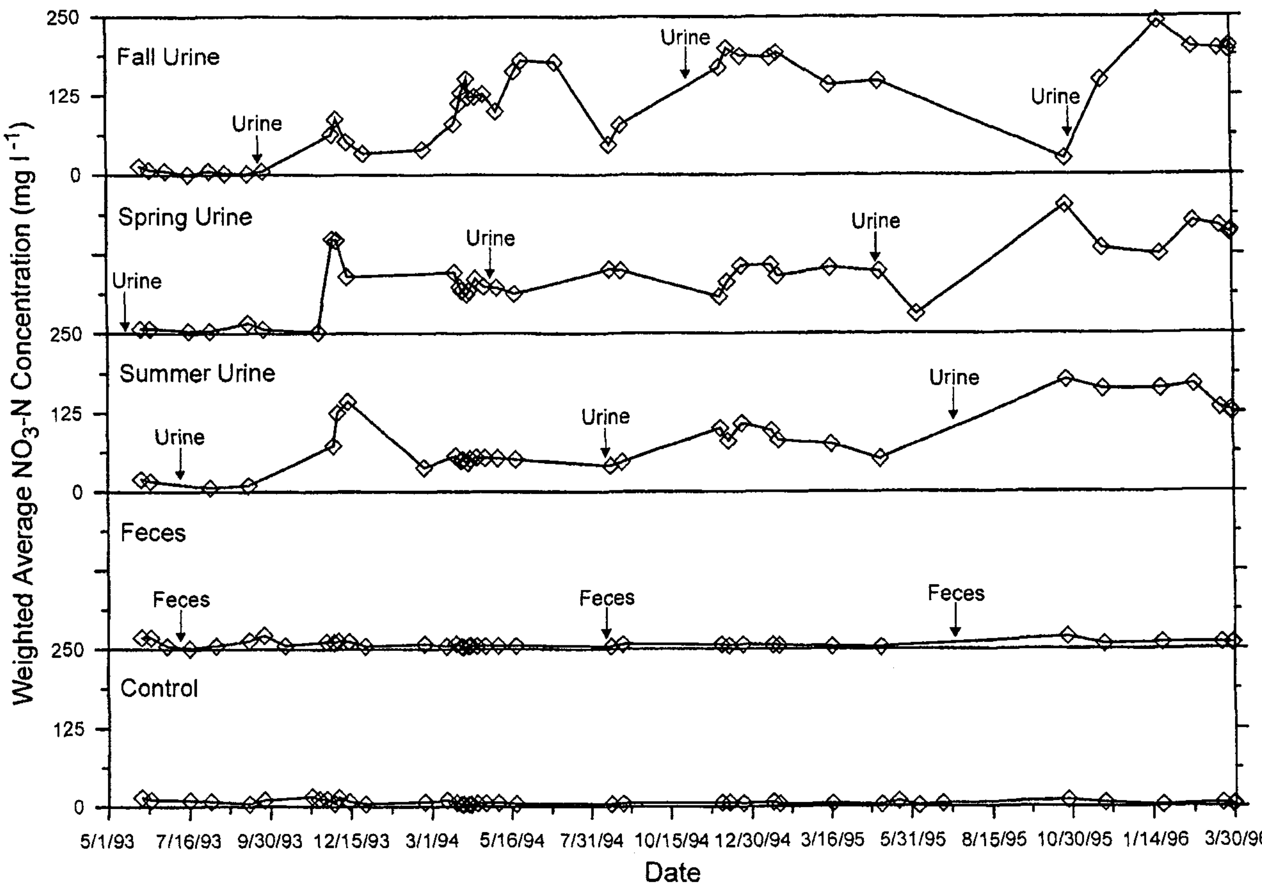

Fig. 3. Nitrate-N concentrations in leachate under spring-, summer-, and fall-applied urine, summer-applied feces, and an untreated control.

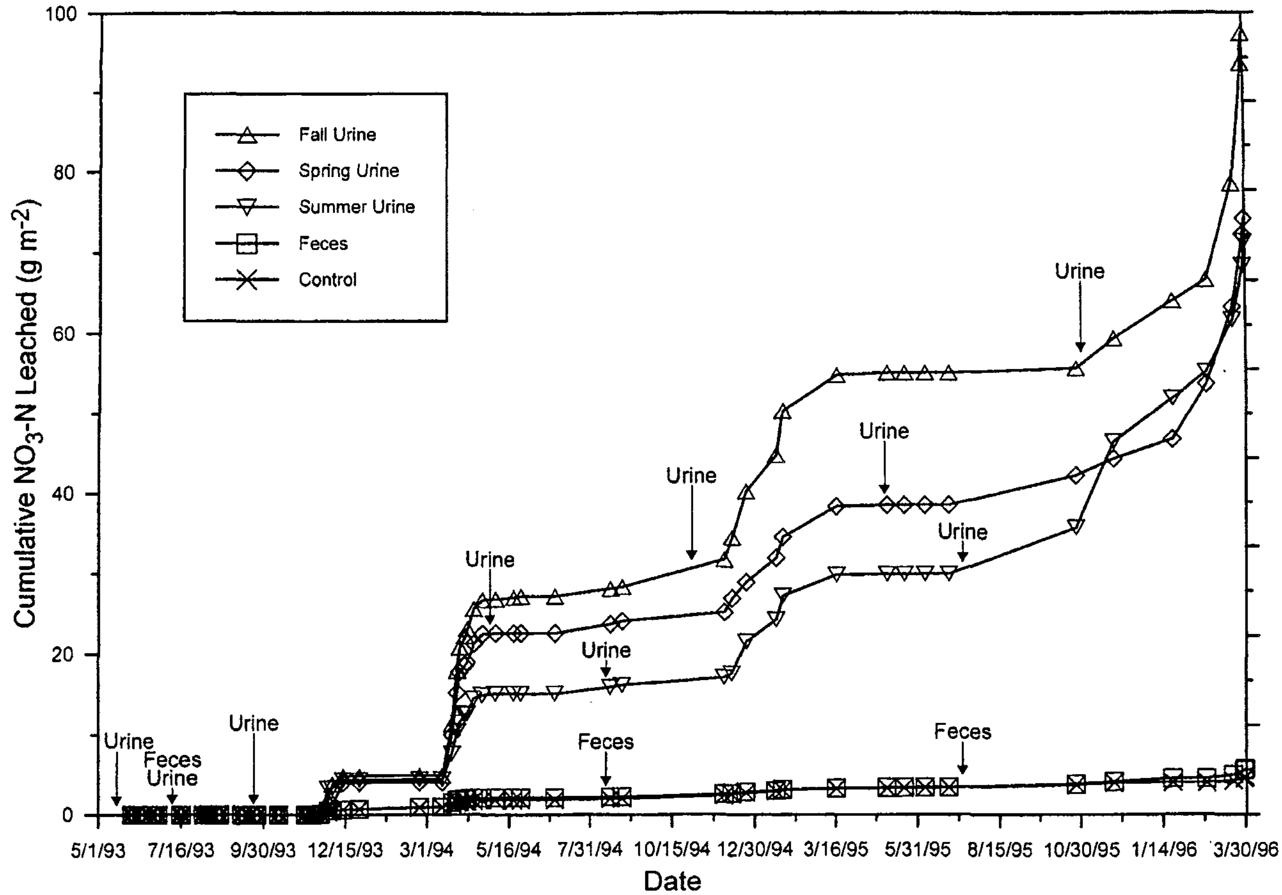

Fig. 4. Cumulative NO_3–N loss beneath spring-, summer-, and fall-applied urine, summer-applied feces, and an untreated control for the water years April 1993 to March 1994, April 1994 to March 1995, and April 1995 to March 1996. The words *urine* and *feces* indicate times of application.

and that the concentration of herbage N in urine spots is a major contributor to these increased losses (Ball et al., 1979) are not new and, as previously stated, have long been a focus of research in the UK, Europe, and New Zealand. Although this research has identified the important mechanisms (leaching, volatilization, and denitrification) by which N is lost from grazed ecosystems, the climatic conditions of the U.S. Northeast are going to determine the relative importance of these mechanisms in this environment.

In a laboratory lysimeter study using synthetic urine conducted under simulated New Zealand climatic conditions, peak leachate NO_3–N concentration from urine was only 42 mg L^{-1} and only 8% of the N applied was lost to leaching (Frazier et al., 1994). This is a sharp contrast to average leachate NO_3–N concentrations of 163 mg L^{-1} beneath urine and the almost 26% N losses that we observed in this study. The temperate oceanic climate (Willy Rudloff, 1981) of this area of New Zealand with a mean winter (June–August) temperature of 6.0°C and a mean summer (December–February) temperature of 16.3°C allowed for year-round cool-season grass growth and uptake of N. In contrast, the temperate continental climate of central Pennsylvania with a mean winter (December–February) temperature of −2.9°C and a mean summer (June–August) temperature of 21.2°C, as measured in our study, only allowed for a growing season of about 7 mo for cool-season grasses. Also, the high winter temperatures and high and consistent water inputs (1600 mm as simulated precipitation + supplemental irrigation) in the New Zealand study allowed for substantial amounts of denitrification. Frazier et al. (1994) estimated that 28% of the N applied as synthetic urine was lost through denitrification, which was attributed to wet soil conditions in their lysimeters. In contrast, water inputs into our lysimeters were less (1071 mm as precipitation) and more erratic (Fig. 1) than in the New Zealand study. At the time of the year when temperatures were the highest (Fig. 1), soil water levels in our study were the lowest as evidenced by the low amount of leachate collected at this time (Fig. 2). Conversely, in the winter when soil water levels and leachate amounts were highest, soil temperatures were lowest. These conditions would not be conducive to denitrification.

Climatic conditions can also interact with the time of year at which urine is deposited to affect the amount of NO_3–N leached. In Ireland and the UK (temperate oceanic climates), it was estimated that only 3% of the urine-N applied from May to September was subject to leaching (Sherwood, 1986; Cuttle and Bourne, 1993). In contrast, 18% of the urine-N applied in May and 28% of the urine-N applied in August was leached in our study. However, when urine was applied in September to November in the Irish and UK studies, about 30 to 66% of the N in the urine was subject to leaching. This is similar to the results from our study in which 31% of the urine-N applied in September through November leached through our lysimeters.

The overall impact of urine deposition on the total amount of N leached from a pasture can be large. Data from the UK (Garwood and Ryden, 1986) show potential NO_3–N leaching was about three times greater under grazing than it was under cut swards of either N-fertilized perennial ryegrass (*Lolium perenne* L.) or white clover (*Trifolium repens* L.) and perennial ryegrass. Similarly, in the Netherlands, 75% of the N leached from grazed grassland was attributed to N returned to the grassland in excreta (Kolenbrander, 1981). Estimates using the data from our study (Stout et al., 1996) indicate that, for a stocking rate of 2.2 mature (682 kg) Holstein cows ha^{-1} d^{-1} for a 180-d grazing season, about 70% of the NO_3–N leached would come from urine deposited on the pasture. This stocking rate would result in an average annual NO_3–N concentrate in the leachate of about 15 mg L^{-1}, a concentration in excess of the 10 mg L^{-1} U.S. primary drinking water standard (U.S. Environmental Protection Agency, 1987). Clearly this indicates that MIG, in and of itself, is not a means of reducing NO_3–N loss from agriculture in the Northeast, and must be considered a livestock production system component that can have negative water quality impacts if not properly managed. One possible management technique would be mechanical harvest and ensilage of late-season herbage growth (Garwood and Ryden, 1986). This would remove animals from the pasture at the time of the year when NO_3–N from their urine would be most subject to leaching.

CONCLUSIONS

As previously observed in the humid maritime climates of the United Kingdom and New Zealand, an appreciable amount of the N excreted as urine in intensively grazed pastures can be leached from the root zone. On a deep, well-drained soil in the temperate continental climate of the northeastern USA, this leaching loss is about 25% of the N contained in the urine. Nitrate-N leaching losses beneath feces amounted to only about 2% of the N in the feces during the relatively short time of this study. Given more time, feces may have more impact on NO_3–N leaching.

If dairy farmers in the Northeast continue to increase the utilization of MIG, the amount of N leached to the groundwater from beneath urine patches could become substantial unless mitigated by improved grazing management. This would be especially true under grazing programs using high N inputs and highly digestible forages, such as ryegrass or brassica species where a larger portion of the N in the pasture is excreted in the urine. Also, grazing programs that involve extension of the grazing season into the fall or involve wintering dry cows on pasture may increase the potential for NO_3–N leaching from urine.

REFERENCES

Ball, P.R., and J.C. Ryden. 1984. Nitrogen relationships in intensively managed temperate grasslands. Plant Soil 76:23–33.

Ball, R., D.R. Keeney, P.W. Theobald, and P. Nes. 1979. Nitrogen balance in urine-affected areas of a New Zealand pasture. Agron. J. 71:309–314.

Barber, H.T., and A.T. Wilson. 1972. Nitrate pollution of groundwater in the Waikato region. J. N.Z. Inst. Chem. 36:179–183.

Barraclough, D., S.C. Jarvis, G.P. Davies, and J. Williams. 1992. The relation between fertilizer nitrogen applications and nitrate leaching from grazed grassland. Soil Use Manage. 8:51–56.

Cuttle, S.P., and P.C. Bourne. 1993. Uptake and leaching of nitrogen from artificial urine applied to grassland on different dates during the growing season. Plant Soil 150:77–86.

Emmick, D.L., and L.F. Toomer. 1991. The economic impact of intensive grazing management on fifteen dairy farms in New York state. p. 19. *In* Forage: A versatile resource. Proc. Am. Forage and Grassl. Counc., Georgetown, TX.

Fales, S.L., S.A. McMurry, and W.T. McSweeny. 1993. The role of pasture in northeastern dairy farming: Historical perspective, trends, and research imperatives for the future. p. 111–131. *In* J.T. Sims (ed.) Agricultural research in the northeastern United States: Critical review and future perspectives. ASA, Madison WI.

Frazier, P.M., K.C. Cameron, and R.R. Sherlock. 1994. Lysimeter study on the fate of nitrogen in animal urine returns to irrigated pasture. Eur. J. Soil Sci. 45:439–447.

Garwood, E.A., and J.C. Ryden. 1986. Nitrate loss through leaching and surface runoff from grassland: Effects of water supply, soil type and management. p. 90–113. *In* H.G. van der Meer et al. (ed.) Nitrogen fluxes in intensive grassland systems. Martinus Nijoff, Dordrecht, the Netherlands.

Harper, L.A., V.R. Catchpoole, R. Davis, and K.L. Weir. 1983a. Ammonia volatilization: Soil, plant, and microclimate effects on diurnal and seasonal fluctuations. Agron. J. 75:212–218.

Harper, L.A., V.R. Catchpoole, and I. Vallis. 1983b. Gaseous ammonia transport in a cattle–pasture system. p. 353–372. *In* R. Lowrance et al. (ed.) Nutrient cycling in agricultural ecosystems. Univ. of Georgia Agric. Exp. Stn. Spec. Publ. 23.

Jarvis, S.C., D.J. Hatch, and D.H. Roberts. 1989. The effects of grassland management on nitrogen losses from grazed swards through ammonia volatilization; the relationship to excretal N returns from cattle. J. Agric. Sci. (Cambridge) 112:205–216.

Kolenbrander, G.J. 1981. Leaching of nitrogen in agriculture. p. 199–216. *In* J.C. Brogen (ed.) Nitrogen loss and surface runoff from land spreading of manures. Martinus Nijhoff/Dr. Junk, The Hague.

Moyer, J.W., L.S. Saporito, and R.J. Janke. 1996. Design, construction and installation of intact soil core lysimeter. Agron. J. 88:253–256.

Parker, W.J., L.D. Muller, and D.R. Buckmaster. 1992. Management and economic implications of intensive grazing on dairy farms in the northeastern states. J. Dairy Sci. 75:2587–2597.

Petersen, R.G., H.L. Lucas, and W.W. Woodhouse, Jr. 1956a. The distribution of excreta by freely grazing cattle and its effect on pasture fertility: I. Excretal distribution. Agron. J. 48:440–444.

Petersen, R.G., H.L. Lucas, and W.W. Woodhouse, Jr. 1956b. The distribution of excreta by freely grazing cattle and its effect on pasture fertility: II. Effect of returned excreta on the residual concentration of some fertilizer elements. Agron. J. 48:444–449.

Ryden, J.C. 1986. Gaseous losses of nitrogen from grassland. p. 37–73. *In* H.G. van der Meer et al. (ed.) Nitrogen fluxes in intensive grassland systems. Martinus Nijoff, Dordrecht, the Netherlands.

Ryden, J.C., P.R. Ball, and E.A. Garwood. 1984. Nitrate leaching from grassland. Nature (London) 311:50–53.

SAS Institute. 1988. SAS/STAT guide for personal computers. Version 6.03 ed. SAS Inst., Cary, NC.

Sherwood, M. 1986. Nitrate leaching following application of slurry and urine to field plots. p. 150–157. *In* A. Dam Kofoed et al. (ed.) Efficient land use of sludge and manure. Elsevier, London.

Shuford, J.W. 1975. Nitrate-nitrogen movement and distribution within a soil profile. Ph.D. diss. Pennsylvania State Univ., University Park, PA (Diss. Abstr. 76-10787).

Steenvoorden, J., H. Fonck, and H.P. Oosterom. 1986. Losses of nitrogen from intensive grassland systems by leaching and surface runoff. p. 85–97. *In* H.G. van der Meer et al. (ed.) Nitrogen fluxes in intensive grassland systems. Martinus Nijoff, Dordrecht, the Netherlands.

Stout, W.L., G.F. Elwinger, S.L. Fales, L.D. Muller, R.R. Schnabel, and W.E. Priddy. 1996. Nitrate leaching from intensively grazed pastures. p. 216–220. *In* Proc. Am. Forage and Grassland Council, Vancouver, BC. 13–15 June 1996. Am. Forage and Grassl. Council, Georgetown, TX.

U.S. Department of Agriculture. 1991. Agricultural statistics. U.S. Gov. Print. Office, Washington, DC.

U.S. Environmental Protection Agency. 1979. Methods for analysis of water and wastes. USEPA, Cincinnati, OH.

U.S. Environmental Protection Agency. 1987. Quality criteria for water. USEPA 440/5-86-001. U.S. Gov. Print. Office, Washington, DC.

Whitehead, D.C. 1995. Nitrogen leaching from soils. *In* D.C. Whitehead (ed.) Grassland nitrogen. CAB Int., Wallingford, UK.

Willy Rudloff, B. 1981. World climates. Wissenschaftlichte Verlagsgesellschaft, Stuttgart.